CHAPTER 5

Candidate Variables for Criteria Setting

A. Introduction

This chapter provides an overview of several trophic state variables that could be used to establish regional and water-body-specific nutrient criteria for lakes and reservoirs. Trophic state variables are those variables that can be used to predict the trophic state of a water body. Trophic state variables include measures of nutrient concentration (e.g., TP, soluble reactive phosphorus, TN, total Kjeldahl nitrogen), plant (macrophyte or algal) biomass (e.g., organic carbon, chlorophyll *a*, Secchi depth), and watershed attributes (e.g., land use). All could be used for establishing criteria to address eutrophication concerns, but only a few are viable candidates for early warning variables. Based on the Proceedings of the National Nutrient Assessment Workshop (U.S. EPA, 1996), the most likely trophic state candidates are TP, TN, chlorophyll, Secchi transparency, and dissolved oxygen. In addition, one watershed metric—land use and the associated phosphorus loading—was recommended as an early warning variable. These variables (metrics or indicators) are briefly reviewed below. EPA presently requires only TP, TN, chlorophyll *a*, and Secchi depth be used, but this set of criteria variables may be augmented by other measurements if the State or Tribe prefers.

Emphasis should be on the open water portion of the ecosystem, and as can be seen in Table 5.1, most of the commonly used biological variables are measures of the amount of organic material in the open water. As discussed later, only a few attempts have been made to incorporate the littoral zone into the assessment of trophic state.

B. Nutrient Variables

1. Phosphorus

Phosphorus and nitrogen are essential nutrients necessary for the growth of plants in lakes. Of these two nutrients, phosphorus is most often considered to be the nutrient that regulates the production of algae in lakes and is most amenable to control. As such, it is often the variable of concern in regards to lake and reservoir eutrophication. Together with algal chlorophyll *a* and Secchi disk transparency, phosphorus is routinely used to estimate trophic status of lakes and reservoirs (see Chapter 2). Vollenweider (1968) and Sawyer (1947) categorized trophic status according to phosphorus concentration. Lakes with phosphorus concentrations below 10 µg/L were classified as oligotrophic; phosphorus concentrations between 10 and 20 µg/L were indicative of mesotrophic lakes; and eutrophic lakes had phosphorus concentrations exceeding 20 µg/L.

Several forms of phosphorus can be measured. Total phosphorus (TP) is a measure of all forms of phosphorus, dissolved or particulate, that are found in a sample. TP has been used throughout North America as a basis for setting trophic state criteria and in developing related models (NALMS, 1992).

Table 5.1. Variables Used to Estimate Trophic State in Lakes

Eutrophication-Related Variables	Apparent Measure	Interference
Total phosphorus	Nutrient concentration, Biomass	Nonbiological, nonalgal forms
Total nitrogen	Nutrient concentration, Biomass	Nonbiological, nonalgal forms
Total organic carbon	Biomass	Nonalgal suspended particulates, dissolved organics
Chlorophyll pigments	Algal biomass, Photosynthetic capacity	Highly variable relationship between chlorophyll and algal carbon or biovolume
Suspended solids	Suspended biomass	Nonalgal particulates
Transparency	Suspended algal biomass	Nonalgal particulates, dissolved color
Turbidity	Suspended algal biomass	Nonalgal particulates
Direct algal counts/ Biovolume	Algal biomass	None, but difficult to do
Biochemical oxygen demand (BOD)	Algal biomass	Nonalgal particulate and dissolved carbon

TP concentrations in runoff or areal exports can be readily related to watershed land use as well (e.g., Reckhow and Simpson, 1980; Walker, 1985a), which makes it an excellent variable for addressing point and nonpoint source loads from the watershed.

Soluble reactive phosphorus (SRP) is a measure of the filterable (filter passings, soluble, inorganic) fraction of phosphorus that is generally thought to be the form directly taken up by plant cells. For this reason, SRP is usually in very low concentrations in lake water, unless phosphorus is not limiting to algal growth. Therefore, it serves more of an indicator of phosphorus limitation than of the trophic status of a lake. For a more complete discussion of the phosphorus forms and their biological significance, see Carlson and Simpson (1996).

TP concentrations vary regionally, as demonstrated by the phosphorus mapping of Omernik (Omernik et al., 1988) and the nutrient ecoregion map in this manual (Chapter 1, Figure 1.3). For example, Minnesota TP concentrations vary substantially between the four ecoregions that contain 98 percent of the State's lakes (Heiskary and Wilson, 1989). Data from Minnesota's ecoregion reference lakes (representative, minimally impacted lakes) demonstrate the variability between and within ecoregions (Table 5.2). The within-region variability can be accounted for in part by the depth of the lakes and mixing status. Table 5.2 reveals the distinct within-region differences in TP as related to lake mixing status. These differences are most pronounced in the north central hardwoods forest lakes where median summer-mean TP concentrations in class I dimictic lakes (mixed only in spring and fall) was 39 µg/L as compared with 89 µg/L for lakes that are continuously mixed (class 3 dimictic or polymictic) lakes. It is likely that internal recycling of phosphorus becomes a significant portion of the phosphorus budget in the shallow eutrophic to hypereutrophic lakes in this ecoregion. Differences are quite

Table 5.2. Median Total Phosphorus (μg/L) Concentrations as Affected by Mixing Status and Ecoregion in Minnesota

Ecoregion	Mixing Status		
	Class I Dimictic (mixes only in spring and fall)	Intermittently Mixed	Continuously Mixed
Northern lakes and forests	20	26	29
North central hardwood forest	39	62	89
Western corn belt plains	69	135	141
Number of lakes evaluated	257	87	199

Source: Heiskary and Wilson, 1988.

pronounced in the western corn belt plains ecoregion as well, but the population of class I dimictic lakes (based on available data) is quite small.

■ **Analysis**

Phosphorus is relatively easy to measure using a colorimetric procedure (APHA, 1998). To use the procedure, all forms of phosphorus must be converted into orthophosphorus. SRP is a form that is defined as filterable and reactive with the molybdate reagent and generally reflects the amount of orthophosphorus plus some polyphosphates in the sample. Acid-washed sample containers and analytical glassware are essential to avoid incidental phosphorus conamination of the analysis.

The distinction between "particulate" and "dissolved" is primarily a function of the filter used. Traditionally, a 0.45μ pore size membrane or GF/F glass fiber filter is used (APHA, 1998). Glass fiber filters are used in some noncritical studies, and micropore filters are used in critical studies. The particulate form of TP is converted to the ortho form with an acid hydrolysis step. The strength and nature of the acid can affect the amount of phosphorus converted. Usually, acid potassium persulfate is used for the digestion (APHA, 1998; Ameel et al., 1993; Ebina et al., 1983; Smart et al., 1981; Menzel and Carwin, 1965).

2. Nitrogen

Nitrogen is also an essential nutrient for algal growth. In contrast to phosphorus, control of nitrogen sources is more difficult because nitrogen can be assimilated directly from the atmosphere by several types of organisms, including some species of the *Cyanophyta*, the blue-green algae. In addition, nitrogen is not as often limiting to plant growth,thus the focus on phosphorus in the majority of eutrophication-related efforts worldwide.

There are several forms of nitrogen to consider, and its cycling is complex compared with phosphorus. The most common forms of concern in eutrophication evaluation are nitrite, nitrate, ammonia, and organic nitrogen, as measured as total Kjeldahl nitrogen (TKN). Total nitrogen (TN) is considered to be the sum of ammonia, nitrate, nitrite, and TKN. Typically nitrate, nitrite, and ammonia are at very low levels in lakes or reservoirs unless there are some relatively recent loadings of manure or fertilizer present in runoff from the watershed or if nitrogen is not limiting to algal growth. These forms are rapidly used by algae and aquatic plants or converted to other forms of nitrogen. The most useful measurement from a modeling standpoint is either TN or TKN. As with TP, TN concentrations vary regionally. Based on data from Minnesota, TN concentrations in the shallow agricultural lakes are about twofold higher than concentrations in the deeper lakes in the forested region.

TN:TP ratios have been used as a basis for estimating which nutrient limits algal growth (e.g., Smith, 1982). Low TN:TP ratios (less than about 7:1) are indicative of nitrogen limitation, whereas ratios greater than 10:1 are increasingly indicative of phosphorus limitation. Based on data from Minnesota, low ratios occur in some shallow hypereutrophic lakes in the northern glaciated plains. However, these low ratios are typically the result of very high TP loads from point or nonpoint sources in the watershed rather than a shortage of nitrogen. Low TN:TP ratios also are found in lakes receiving significant amounts of sewage effluent.

■ **Analysis**

Like phosphorus, nitrogen is divided into dissolved and particulate forms based on whether or not a particle passes through a filter. It would be advisable to use the same size filter for both phosphorus and nitrogen. TN is similar in concept to TP, being the estimate of all nitrogen forms. Traditionally, TN is calculated as the sum of the analyses of all nitrogen forms ($NO_3^- + NO_2^- + TKN^-$). Newer tests allow the conversion of all forms to NO_3 and are therefore direct equivalents to the TP test except alkaline persulfate digestion is used (APHA, 1998; Solorzane and Sharp, 1980; D'Elia et al., 1977).

C. Biological Variables

1. Organic Carbon

The term "biomass," used so frequently in the ecological literature, refers to the weight of living material in a unit of measure (in a bacterium, in a cubic meter of water, or in an ecosystem). The bulk of that weight is in the form of organic carbon. Organic carbon production or productivity (the rate at which carbon is fixed in the aquatic ecosystem) has been the basis for numerous trophic state classification systems and for the definition of trophic state itself (Rodhe, 1969). The rates of production and decomposition of carbon compounds and the resulting biomass are at the heart of the eutrophication problem.

Despite the central character of carbon in eutrophication and ecosystem structure and function, carbon has not been explicitly measured or modeled in most standard eutrophication or nutrient/food chain frameworks. This omission may have been, in large part, the result of the difficulty in measuring and interpreting organic carbon. Carbon analysis requires expensive, dedicated equipment, and each type of instrument produces slightly but significantly different estimates of carbon. Although the criticism of technique-specific results can probably be invoked for all of the variables discussed in this chapter, the expense of the equipment plus the variety of techniques available may have restricted the popularity of the routine analysis of carbon.

A second reason for the limited use of carbon in eutrophication-related studies is that chlorophyll pigments or direct measures of algae and macrophytes not only estimate plant biomass, but also directly indicate the photosynthetic capacity to produce carbon. Chlorophyll also remains the only economic means to directly measure algal biomass free from significant interferences. However, the chlorophyll-to-carbon ratio in phytoplankton varies by about a factor of 5 depending on ambient light and nutrient levels (Laws and Chalup, 1990). Consequently, measuring particulate organic carbon together with chlorophyll could better assess the amount of autochthonous carbon, particularly during bloom conditions.

There are several reasons for measuring organic carbon. Organic carbon now can be measured more easily and precisely than in the past. The use of carbon associated with algal metabolism and decomposition greatly facilitates the modeling of dissolved oxygen. Because many shallower stratified systems experience hypolimnetic anoxia, models incorporating the carbon factor must be capable of accurately simulating this phenomenon. The direct modeling of organic carbon becomes especially important for systems where both allochthonous and autochthonous carbon sources are important. The use of eutrophication models as the basis for examining the transport and fate of toxic substances in lakes and impounds requires that the amount and forms of organic carbon be specified. Such confounding problems include toxic organics, metals, and disinfection byproducts. Finally, the state of the lake's bottom sediments is inextricably tied to the amount of carbon it receives from the overlying waters. This has ramifications for sediment oxygen and toxic substance sequestering.

2. Chlorophyll a

Chlorophyll is the major photosynthetic pigment in plants, both algae and macrophytes. As such, it is the important variable when any estimate of the photosynthetic capacity of an ecosystem is desired. Chlorophyll is probably most often used as an estimator of algal biomass. The relationship between chlorophyll and TP is well established for lakes and reservoirs across much of the world. Despite the curious fact that the chlorophyll molecule itself contains no phosphorus while TP includes phosphorus dissolved in the water as well as in algal cells, relationships between TP and chlorophyll have dominated the empirical linkages between nutrients and the biological response of the algae in lakes. See Nurnberg (1996) for a recent review of these relationships.

Chlorophyll is also a preferred variable because there are lakes where TP is not the sole or primary limiter of algal production or biomass, for example, lakes with high inorganic turbidity or high flushing rates. For instance, a chlorophyll a goal of less than 30 µg/L was used for Lake Pepin, a run-of-the-river reservoir on the Mississippi River between Minnesota and Wisconsin (Heiskary and Walker, 1995). In this reservoir, inorganic turbidity and high flushing rates were the primary factors controlling algal production during above-average flows (about 20,000 cfs or greater), and chlorophyll a routinely remained below 30 µg/L at these flows. In contrast, as flows declined below about 20,000 cfs and residence time increased above 10 to 14 days, chlorophyll a increased as the influence of residence time and inorganic turbidity declined, and the potential trophic status, as reflected by TP, was realized.

Because of the relationship between chlorophyll and phosphorus and its linkage to algae biomass, chlorophyll is often a major component of trophic state indices (Carlson, 1977) and water quality criteria. Oregon has set an endpoint of 10 µg/L for natural lakes that thermally stratify and 15 µg/L for natural lakes that do not thermally stratify (NALMS, 1992). Similarly, North Carolina uses a standard of 40 µg/L for warm waters and 15 µg/L for cold waters (NALMS, 1992). On the regional level, Raschke (1994) proposed a mean growing season limit of 15 µg/L for water supply impounds in the

southeastern United States and a value of 25 µg/L for water bodies primarily used for other purposes (e.g., viewing pleasure, safe swimming, fishing, boating).

In addition to the use of chlorophyll in classification, the chlorophyll interval frequency (bloom frequency) can be predicted based on regression equations developed by Walker (1985b) relating blooms to phosphorus. These chlorophyll *a* intervals can be related to varying user perceptions of lake condition. The projected frequency of these extreme events, as a result of increased phosphorus loading can be readily understood by citizens and decisionmakers (Heiskary and Walker, 1988).

■ **Analysis**

The term "chlorophyll" really represents a family of molecules, chlorophyll *a*, *b*, *c*, and *d*. Chlorophyll *a*, because of its primary role in photosynthesis, is often the molecule of preference. Chlorophyll can be measured by several different methods. Most include steps of concentration of the algae, extraction of the chlorophyll with a solvent, and measurement of the chlorophyll molecule. Unfortunately, each of these techniques and choices of solvents, extraction times, and methods for measuring the chlorophyll can produce widely different estimates of chlorophyll.

Chlorophyll *a* is the primary photosynthetic pigment and can be measured quite accurately with high-performance liquid chromatography (HPLC) analysis. The use of colorimetric techniques to measure chlorophyll *a* free from all forms of interference from either decomposition products (phaeophytin, phaeophorbides, or chlorophyllides) or other chlorophylls (chlorophylls *b* and *c*) is almost impossible. However, if the purpose of measuring chlorophyll is to estimate biomass rather than to know the absolute amount of photosynthetic pigment, the absolute accuracy obtained by HPLC is not necessary and the spectrophotometric techniques suffice. Most published relationships between TP and chlorophyll were made using these cruder spectrophotometric methods.

Because chlorophyll *a* cannot be measured without interference, Carlson and Simpson (1996) recommend using total chlorophyll pigments (the estimated chlorophyll at a single wavelength) rather than subjecting the sample to further manipulations to obtain an estimate of chlorophyll *a*. However, the acidification step recommended in the Standard Methods (APHA, 1998) does remove some phaeophyton interferences, and the resultant chlorophyll concentration measured may be closer to the actual concentration than an unacidified measure. Fluorometry also can be used, especially if algal densities are very low and a very sensitive method of detection is needed.

Chlorophyll pigments degrade easily. Whole nonfiltered samples can apparently be kept for several days if left in the dark and in the refrigerator. Filtered samples should be kept dark and frozen. Filtered samples immersed in the solvent are apparently stable as long as they are kept dark. More information on chlorophyll preservation can be obtained from Carlson and Simpson (1996) and the American Public Health Association (1998).

The important consideration for any of these techniques is that standardization of methodology is critical. Alteration of the extraction solvent, extraction time, type of filter used to concentrate the sample, or concentration of acid used to convert chlorophyll into phaeophytin all can alter the concentration estimates. Therefore, once a technique is chosen, for the sake of data consistency, the technique should be altered only with the knowledge that some of the previously collected data may not be compatible. This is one reason for using total chlorophyll; it requires the least analytical manipulation and therefore is the most conservative estimator of chlorophyll. Caution also should be used when obtaining data from several sources that use different techniques that may produce discrepancies. This is

particularly important when compiling several data sets to establish reference conditions for criteria development.

3. Secchi Disk Transparency

Secchi disk transparency, or Secchi depth, can cheaply provide a great deal of information on lake water quality and, together with TP and chlorophyll *a*, has become routinely used as a measure of lake trophic status (e.g., Carlson, 1977). Secchi depth is routinely incorporated into citizen volunteer monitoring programs, and in many States, it often provides the best basis for identifying trends in trophic status over time (Heiskary et al., 1994). Smeltzer et al. (1989) found Secchi transparency to be the best variable for identifying statistically significant trends in trophic status because of the large number of observations that can be gathered in a given season and the ability to gather numerous years of data at little or no cost as compared with TP and chlorophyll *a,* which are typically monitored at a much reduced frequency and at a higher cost. As with other variables, Secchi depth may vary considerably in a given lake between and within seasons; therefore, it is desirable to have an "indicator season." In many States, the most reliable timeframe for measuring Secchi transparency and estimating lake trophic status is summer, typically mid-June to mid-September. Thus, this is the focus for much of the citizen and professional data gathering that takes place. Summer-mean measures of TP, chlorophyll *a*, and Secchi depth then are used to estimate lake trophic status. It should be noted that Secchi depth measurements are inadequate for nutrient level estimations in lakes with colored water or inorganic suspended solids. In such instances, the TN and TP measurements are more telling indicators of enrichment.

User perception measurements may be taken in conjunction with Secchi readings. These user perceptions, especially when recorded by citizen volunteer monitors, can provide a good basis for associating designated uses and subjective perceptions of water quality with actual measurements of water quality. Citizen volunteer programs in Minnesota, Vermont, and New York routinely collect this information. Smeltzer and Heiskary (1990) found that user perceptions may vary between States and may vary further between regions within a given State. This type of information can provide a perspective useful for criteria and goal setting purposes (see Vermont and Minnesota case studies in Appendix B).

■ **Analysis**

The standard Secchi disk used in limnological investigations is a 20 cm diameter disk that either is all white or has alternating black and white quadrants. Techniques vary as to how the depth should be measured, and statistically significant differences between the techniques have been reported. Probably the best method is to lower the disk on the sunlight side of the boat to eliminate shading the disk or to have the disk disappear in a background darkened by the shadow of the boat. Glare on the water is eliminated by using a viewscope to view the disk. The disk is lowered until it cannot be seen, the depth is noted, and then the disk is raised until it can be seen again. The average between the depth of disappearance and the depth of appearance is called the Secchi depth.

Considerable variations of this technique exist. Probably more programs lower the disk on the shady side of the boat and do not use a viewscope. The use of a viewscope or the side of lowering will give different results. The choice of an all-white or a black-and- white disk also will give different readings, especially in clearer waters. The point to be emphasized is that the particular method chosen may not be as important as consistency in the method of Secchi depth determination. Data consistency almost dictates that the technique cannot be changed without transforming part of the data record or losing

historical information. When consistently applied in conjunction with TP and TN calibrations, Secchi depth change is a helpful tool for blue-green algal bloom prediction in north temperate lakes.

4. Dissolved Oxygen

Dissolved oxygen (DO) and temperature profiles are routinely taken in eutrophication-related studies. These measurements are essential for characterizing the mixing status and for determining the presence or absence of oxygen above the sediments, the rate of hypolimnetic oxygen depletion, the number of days of anoxia, and whether the lake has suitable habitat for sensitive fish species. Anoxic conditions in lakes also may favor the growth of blue-green algae such as *Microcystis* (Reynolds and Walsby, 1975). The lack of oxygen in the bottom waters causes sediments to release such dissolved constituents as inorganic phosphorus, ammonia, and hydrogen sulfide. The initial disappearance of oxygen in the hypolimnion can occur before any noticeable change in the productivity of algae in the epilimnion because of the amplification of organic epilimnetic inputs by the sediments (Gliwicz and Kowalczewski, 1981). This makes the oxygen content of the hypolimnion and the rate of disappearance to be a potential early warning system of changes in trophic state.

Oxygen concentrations and rates of depletion have been used to characterize lakes and can in some instances be related back to nutrient status. In contrast to the previous variables where epilimnetic or composite measurements were key, the focus for DO in lakes is primarily on hypolimnetic concentrations. DO concentrations in the hypolimnion can be related to epilimnetic TP and annual primary production and inversely to mean summer Secchi depth (Cornett and Rigler, 1979). Checking for the presence or absence of hypolimnetic oxygen was an early method of discriminating between oligotrophic and eutrophic lakes (Thienemann, 1921) that is still used to some extent today. Hypolimnetic oxygen depletion rates have been used as a variable of trophic state by several investigators (Mortimer, 1941; Rast et al., 1983). This would not apply to allochthonous organic loading, which may deplete DO independent of nutrient concentrations.

Because the presence or absence of hypolimnetic oxygen and oxygen depletion rates are confounded by the size of the hypolimnion, the rate usually is indexed to the area of the hypolimnetic surface and termed the areal hypolimnetic oxygen deficit (AHOD) (see Hutchinson, 1957; Wetzel, 1975). Compensating for the size of the hypolimnion will not necessarily give a value correlated with the amount of productivity or nutrient status of the epilimnion because the rate of oxygen consumption is dependent on temperature as well. Latitudinal differences in AHOD will exist independent of the trophic status (biomass concentration) of the lake. The hypolimnetic oxygen depletion rate is also affected by dissolved color; Hutchinson (1957) recommends that AHOD not be used on lakes with color greater than 10 Pt units.

Walker (1979) used trophic status, AHOD, mean hypolimnion depth, and oxygen concentration at spring turnover to predict the effective number of days of oxygen supply present in the hypolimnion after spring turnover. Nurnberg (1996) quantified hypolimnetic anoxia based on DO profiles and lake morphometry and related this anoxic factor to other trophic status variables. Thus, DO is an important variable to consider when assessing the impact of eutrophication, even though it does not define trophic state.

■ Analysis

Measurements are typically taken at appropriate intervals from the surface to the bottom of the lake on each sample date. Oxygen is typically taken using a remote sensing probe, although oxygen also can

be measured by analyzing individual samples using the Azide Modification of the Winkler technique (APHA, 1995). The minimum frequency for characterizing mixing and oxygen status of the lake is dependent on the rate at which oxygen can be depleted in the water body, which itself is dependent on the size and temperature of the hypolimnion and the potential amount of organic matter that may be settling into this region. In some cases, the minimum frequency may be a month; in others, it may only be a few hours. Some shallow lakes experience daily oxygen depletion near the bottom.

5. *Macrophytes*

The term "macrophyte" refers to any plant life larger than the microscopic algae in aquatic systems. It may be a plant rooted in the sediment, such as pond weeds or cattails, or that is free-floating, such as duckweed or coontail. It also includes large algae such as *Chara*. Macrophytes are important in any consideration of trophic state because they are also plants and, therefore, are potential utilizers of incoming plant nutrients. Although a great deal of research has been done on macrophytes, the ability to predict the extent of macrophytes based on nutrient load or even nutrient concentration still has not been attained.

Although macrophytes require nutrients for growth, the immediate origin of those nutrients is still a matter of some controversy. Originally it was thought that macrophytes may compete with algae for nutrients in the water; this may be the case for floating species, such as duckweed or coontail. Evidence suggests that rooted aquatic plants draw most, if not all, of their nutrients from the sediments, not from the water. In this manner, they can obtain nutrients from the sedimented historical phosphorus. This use of historical nutrients obscures or even eliminates correlations and predictions of macrophyte biomass based on contemporary nutrient loading. It also discourages the management of macrophyte-based eutrophication by nutrient loading control because the macrophytes may persist and most likely even spread after nutrient reduction.

Despite the lack of correlation between macrophyte biomass and nutrient loading, they still may be related. Nutrients attached to particles will settle out, bringing new substrate and nutrients to the macrophytes. Dead algae will also settle, again increasing substrate and nutrients. The presence of macrophytes near water inputs may actually serve to intercept sediment particles, thus building up these regions faster than if the particles were allowed to sediment throughout the lake. Increased nutrient loading can be expected to enhance the sedimentation rate and thus increase the areal coverage of macrophytes (Carpenter and Lodge, 1986). The macrophytes may actually serve as a positive feedback mechanism that enhances the filling in of lakes, their own structures serving to fill in the littoral area and increasing the colonizable area even more.

Considerable evidence suggests that some sort of antagonism exists between macrophytes and floating algae in lakes and ponds. Macrophyte density may be suppressed when algal densities are high, presumably because the algae and/or epiphytes on plant surfaces shade out macrophytes. Conversely, if macrophytes are dense, there are accounts of the algae not growing, even in the open areas outside the macrophyte beds. It has been hypothesized that a chemical is excreted by the macrophyte that suppresses algal growth. According to the most sweeping theory relating eutrophication to the relationship between algae and macrophytes, as a lake eutrophies it may become dominated by either algae or macrophytes, but not both (Scheffer, 1989; Scheffer et al., 1993). Catastrophic events, such as herbiciding or weed harvesting, can shift a macrophyte-based system entirely into an algal-based system. Alternatively, killing the algae by algicides or by nutrient reduction will shift the system into a macrophyte-dominated system with little algae present. Both of these systems are stable and will persist unless again shifted by external events. If this model is correct, assessment and prediction of the response to nutrient loading

increases or reductions will be difficult unless the mechanism underlying these alternate states is known. However, the ability to manipulate the system as Scheffer (1989) has suggested allows the selection of the most desirable type of condition, algae or macrophytes, for a given use and enrichment level.

Most trophic state variables and, indeed, all simple nutrient loading models ignore the growth and extent of macrophytes, probably reflecting the early limnological emphasis on open-water algae. This neglect of the macrophytes hinders the determination of the impact of nutrients on the entire biological system. If nutrient loading models or trophic state variables do not account for macrophyte biomass, they may underestimate the potential impact of nutrients in macrophyte-dominated lakes.

■ **Analysis**

The solution for macrophyte biomass determination could be very simple. Canfield et al. (1983) proposed the assessment of trophic state based on the TP concentration of the lake, including the amount of phosphorus in macrophytes in the lake. The total macrophyte biomass in the lake (kg) is estimated by the equation:

$$TSMB = SA \times C \times B$$

where TSMB = total submersed macrophyte biomass, SA = lake surface area, C = percent cover of submersed aquatic macrophytes, and B = average biomass collected with a sampler.

Canfield et al. (1983) estimated the TP in plant biomass based on the phosphorus in each species and the relative abundance of each species. The TP content of the lake was obtained by adding the amount of phosphorus in the macrophytes to the amount estimated to be in the water column. There seems to be no reason why the same approach could not be used to measure total plant biomass or chlorophyll. Trophic state could then properly include both macrophytes and algae and would have internally consistent units. This would be an instantaneous measure, as sediment phosphorus availability to the biota is assumed to be measured in the macrophytes and algae.

6. Biological Community Structure

Some early trophic classifications were based on the biota of lakes. Thienemann (1921) simultaneously developed a classification scheme based on the species of benthic chironomids in lakes and on the hypolimnetic oxygen concentration that affected their species composition. It must have seemed reasonable at the time that the classifications of Thienemann (1921) and Naumann (1929) could be joined, because Naumann's eutrophic lakes also lacked oxygen in the hypolimnion and had distinct benthic fauna (Thienemann, 1921).

Because of Thienemann, the benthos of lakes has received a great deal of attention in trophic classification. The array of Chironomids, clams, and oligochaete worms have been shown to change with trophic state. In most cases, these changes are related to the loss of oxygen in the hypolimnion. Algal species change as a lake becomes more eutrophic, with a dominance of diatoms shifting to cyanobacteria (blue-green algae). Certain species of diatoms are characterized as being found largely in oligotrophic lakes while others are found in eutrophic situations. Because the frustules of diatoms are preserved in the sediment, this change in species allows a paleolimnological investigation of past trophic states. Zooplankton species change as well, but their change is confounded by alterations in the intensity of

predation upon them as the density of zooplanktivorous fish increases, perhaps the result of alterations in the density of the macrophytes that give the fish shelter from their predators.

In general, fish yield increases as the productivity of the lake increases. However, there may be changes in the dominant fish species as a lake eutrophies (Oglesby et al., 1987) (Figure 5.1). In northern lakes, salmonids may dominate in clear lakes having oxygenated hypolimnia. When primary productivity increases to the point that the hypolimnion becomes anoxic, salmonids may disappear to be replaced by percids; percids then are replaced by centrarchids; and finally, at the highest nutrient concentrations, rough fish such as carp or bullheads prevail.
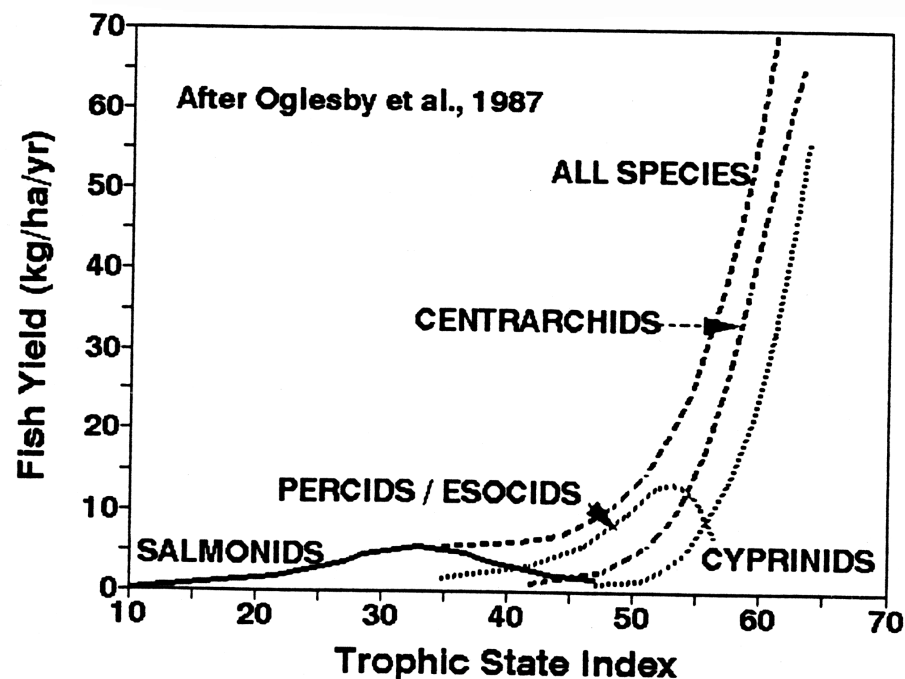
Figure 5.1. Changes in fish species yield with trophic state.

■ **Analysis**

Unfortunately, changes in biological structure do not fit neatly into a nutrient-based classification because structural changes can occur along any environmental axis such as pH or temperature. The bioassessment of aquatic habitats has its strength in the concept that the organisms can be sensitive variables of the condition of the aquatic environment. However, unless a great deal is known about the requirements of the organisms themselves, the assessment does not necessarily indicate the nature of the disturbance. Such general variables would be of little use as variables of nutrient change if they were susceptible to change by a large number of other factors as well.

The fact that the assessment of the species or species complexes in a lake may not be the sole indicator of nutrient-related changes does not mean that assessing and tracking the biological structure would not be useful. It could be that subtle changes brought about by nutrient enrichment may affect one or more groups, and these changes may be more apparent in the structure than in the biomass.

D. Land Use

Changes in land use in a lake's watershed is a viable early warning indicator of potential lake eutrophication. Landscape data can be used to characterize the watersheds of a population of lakes and to estimate potential nutrient loadings under a variety of management scenarios. For example, the State of Maine seeks to protect lake water quality by limiting the acceptable increase in phosphorus in their lakes that may result from changes in land use in the watershed. Very strict requirements for handling storm water are a primary aspect of their approach (see Maine case study in Appendix B). The changes in land use of most concern are typically the shift from forested or open uses to agricultural or urban land uses. Phosphorus exports and concentrations associated with various land uses are fairly well documented in the literature (e.g., Reckhow and Simpson, 1980) as well as the effects of the increased loading on lakes. Thus, increases in phosphorus loading as related to changes in land use can be estimated, and the impact of the changes can be described by means of empirical models. Whenever possible, it is advisable to use phosphorus concentration and export data summarized based on watersheds in a State or region because phosphorus export may vary between regions as well as between land use types. Some general notes and examples follow.

The generalized land use categories often considered for modeling or prediction purposes are forest, water and marsh, cultivated, pasture/open, and developed (urban and residential). While phosphorus exports and concentrations from a given land use may vary substantially in the literature, some general patterns emerge. The average phosphorus concentrations found by Omernik (Rohm et al., 1995) give some indication of the radical changes in concentration alone when land is disturbed. If the changes in water loading caused by increased impervious surfaces and decreased plant transpiration are included, the potential impact of human activity in the watershed can be easily seen.

Phosphorus exports from forested lands are typically low, on the order of 0.1 to 0.15 kg phosphorus/ha/yr (Reckhow and Simpson, 1980; Verry and Timmons, 1982). Based on data from the predominantly forested Northern Lakes and Forest ecoregion of Minnesota (Table 5.2), stream TP concentrations typically range from 20 to 50 µg/L (McCollor and Heiskary, 1993) (Table 5.3). This range of exports and concentrations is often applicable for marsh land use as well, although phosphorus export will vary seasonally in marshes.

Pastured and open land use is a somewhat nebulous category that might include idle grasslands (e.g., Conservation Reserve Program, [CRP]), park lands, or heavily pastured lands. Feedlots should not be

included in this category but rather should be considered separately with estimates made on a per animal unit basis. Pastured and open park land exports often range from about 0.2 to 0.4 kg phosphorus/ha/yr. For example, two monitored subwatersheds in southwest Minnesota, with 60 percent or more of the watershed in CRP, had phosphorus exports of 0.25 to 0.40 kg phosphorus/ha/yr (Schueler, 1995).

Phosphorus export from cultivated lands is frequently high and variable. Reckhow and Simpson (1980) note that phosphorus export might vary between mixed agriculture (0.4 to 2.3 kg phosphorus/ha/yr) and row crops (0.2 to 0.9 kg phosphorus/ha/yr). Phosphorus exports from two southwest Minnesota watersheds characterized by 81 percent and 49 percent cultivated land use were 0.4 and 0.6 kg phosphorus/ha/yr, respectively. Phosphorus concentrations from streams in the highly agricultural western corn belt plains ecoregion of Minnesota typically range from 160 to 330 µg/L (Table 5.3). Prairie and Kalff (1986) suggest however that phosphorus export from agricultural land varies as a function of watershed size, and they present equations for calculation of phosphorus export by land use type and watershed size. In general, as watershed size increases, phosphorus export tends to decrease in agricultural lands, with row crops and pasture exhibiting the greatest decrease. In practice, this often leads to phosphorus export coefficients on the order of 0.2 to 0.6 kg phosphorus/ha/yr for cultivated lands. This was not the case in forested watersheds, where little change, as a function of watershed size, was noted.

Urban land uses tend to export phosphorus at rates often equivalent to or higher than some cultivated land uses. The extent of impervious surfaces is a primary reason for higher export rates. These impervious surfaces are very efficient conduits for exporting water and contaminants off the landscape.

Table 5.3. Interquartile Range of Phosphorus Concentrations (µg/L) for Minimally Impacted Streams in Minnesota by Ecoregion, 1970-1992

Ecoregion	Percentile		
	25%	50%	75%
Northern lakes and forests	20	40	50
Northern Minnesota wetlands	40	60	90
North central hardwood forest	60	90	150
Northern glaciated plains	90	160	250
Red River valley	110	190	300
Western corn belt plains	160	240	330

Source: McCollor and Heiskary, 1993.

Thus, high phosphorus exports from urban land uses might be more a function of efficiency of delivery rather than land use per se. Reckhow and Simpson (1980) suggest a range of 0.5 to 1.25 kg phosphorus/ha/yr for urban land uses. Walker (1985a) estimated urban phosphorus export of 0.5 kg phosphorus/ha/yr for low-density residential use in Minnesota and 1.2 kg phosphorus/ha/yr for mixed urban and commercial use. The higher range of phosphorus exports might be appropriate where storm sewers drain impervious areas without the benefit of intervening sedimentation basins. Bannerman et al. (1993) in a comprehensive study of storm water in Madison, Wisconsin, found TP concentrations, as monitored at specific sites in the city, to range from a low of 150 µg/L from roofs to 2,670 µg/L from lawns. However, in terms of critical-source areas and contaminant-load percentages, streets and driveways, where runoff volumes were high, accounted for 78 percent of the overall TP loading from the residential land use area, while lawns, where runoff volumes were low, accounted for 14 percent in their study.

■ **Analysis**

The availability of land use data may vary between locales. However, with the increasing use of Geographic Information System (GIS) databases, this aspect of lake and watershed assessment should become easier in the future. In general, the first step is to delineate the total watershed of the lake. It may be worthwhile to also delineate the immediate watershed (that portion which drains directly to the lake without going through another lake or major wetland) because land use changes in that portion of the watershed may ultimately have the greatest impact on the lake of concern. Once the watershed is delineated, land use information might be acquired from GIS data, aerial photos, and/or other records that might be available through soil and water conservation districts, local planning and zoning offices, or other sources. Land use categories should be mapped, and summaries in terms of the total area (e.g., hectares) and percent composition of the watershed by land use type should be noted. This information combined with current monitoring data should provide a basis to begin evaluating the effect of future changes in land use on the water quality of a lake. It also may be instructive to have land use data from a representative subset of lake watersheds in a given State or ecoregion for comparison purposes. For example, the land use composition of reference lake watersheds has been evaluated by ecoregion for Minnesota (Table 5.4). This provides a basis for comparing the land use assemblage for the lake under study compared with the typical composition for lakes in the same region. This comparison may, in part, explain deviations in water quality from regional norms or typical values.

Table 5.4. Typical Watershed Land Use Composition for Minnesota Ecoregion Reference Lakes (based on interquartile range)

Land Use (%)	Northern Lakes Forests	North Central Hardwood Forests	Western Corn Belt Plains	Northern Glaciated Plains
Forest	54-81	6-25	0-15	0-1
Water and marsh	14-31	14-30	3-26	8-26
Cultivated	<1	22-50	42-75	60-82
Pastured	0-6	11-25	0-7	5-15
Cultivated and pastured	0-7	36-68	48-76	68-90
Developed	0-7	2-9	0-16	0-2